LCA Case Studies

Uncertainty in Life Cycle Impact Assessment of Toxic Releases

Practical Experiences – Arguments for a Reductionalistic Approach?

Impact Assessment of Toxic Releases in a Substance Flow Analysis for PVC in Sweden

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Abstract

This paper describes the experience with impact assessment of toxic releases in a Substance Flow Analysis (SFA) for PVC in Sweden. For this system, all emissions related to the PVC-chain were inventoried. They have been evaluated making use of the Life Cycle Impact Assessment (LCIA) step from the CML-guide, including the new toxicity equivalency factors calculated with the Uniform System for Evaluation of Substances (USES). The application of this method led to the conclusion that LCA Impact Assessment of toxic releases is still a major bottleneck: the USES-equivalency factors are not to be trusted due to outdated data, inappropriate defaults, etc. in the USES' substance properties database. Therefore, a second USES-set of factors was calculated that differed up to factors of 1,000 or more from the old ones. Even these factors probably suffer from unacceptable high structural, in practice not reducible uncertainties. In conclusion, we warn the LCA community not to overestimate the possibility of LCA Impact Assessment to obtain a meaningful priority setting with regard to toxicity problems. Instead, we propose developing indicator systems for LCIA of toxic releases that genuinely deal with all relevant types of uncertainty: data uncertainty, modelling uncertainty and particularly paradigmatic uncertainty.

Keywords: LCA methodology, uncertainty, PVC; PVC, uncertainty, LCA; uncertainty, PVC, LCA; USES, uncertainty, PVC

1 Introduction

During the last decade, Life Cycle Assessment (LCA) has developed itself into an important tool for the environmental improvement of products and processes. Starting as a methodology mainly focused at Life Cycle Inventory (LCI), concerning mass balance accounting of energy and materials of a complete system, large efforts have been made since the late eighties to use LCI data towards examining other environmental aspects of a system. Proposals for structuring this so-called Life Cycle Impact Assessment (LCIA) have been made among others by a number of international bodies and in a number of (inter)national methodology development projects (HEIJUNGS et al., 1992; CONSOLI et al., 1993; LINDFORS et al., 1995; BRAUNSCHWEIG et al., 1996). In many of those documents the desire is expressed that the list of environmental problems is as complete as possible, and that a high level of sophistication is striven for with regard to the way inventory data are used in the impact assessment (e.g. where possible making use of fate, effect and sometimes also background level and spatial information). This direction has been termed the expansionistic approach to LCA. At the same time, there have been warnings that striving for too much sophistication and completeness could not be justified due to inherent weaknesses and uncertainties in the tools currently available for LCIA (OWENS, 1996). Advocates of this view propose limiting the impact assessment in LCA to a small number of robust indicators, or to perform no impact assessment at all. This direction has been called the reductionalistic approach to LCA (KLÖPFFER, 1996).

Impact assessment of toxic releases is one of the most complex areas in LCIA. It thus may be the field for which this discussion is most relevant. TNO and CML experienced the practical usefulness of various toxicity impact assessment methods in recent case studies on chlorine and PVC that were dominated by tense societal debates on toxicological aspects (KLEIJN et al., 1997; TUKKER et al., 1995; 1996; 1997 and 1998). Taking specifically the PVC-case as a basis, this paper addresses the following points:

- practical experience with uncertainties in the USES-based method for toxicity impact assessment in LCA in the case of PVC;
- an analysis of the probable importance of these uncertainties on a more structural, case-independent level, and their consequences for the viability of LCA impact assessment of toxic releases;
- a review how such uncertainties have been addressed in the past, and an explanation for the apparent under-estimation of such uncertainties;
- recommendations for approaches to address uncertainties adequately in LCA impact assessment of toxic releases.

Though this paper primarily reflects experiences with regard to LCIA of toxic releases, we feel that there are parallels to other impact categories. The outlook in the final section may therefore have broader implications than for LCIA of toxic releases alone.

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2 Life Cycle Impact Assessment of Toxic Releases

Over the years, more and more sophisticated impact assessment methods have been developed for toxic releases. Around 1990, the so-called critical volumes approach was often used. A quantity of emitted substance to air was divided (mostly) by a policy standard for the desired air or water quality. Since those policy standards were not based on a uniform toxicological basis, subjective elements in the toxicity score of a substance were introduced. The CML-guide for LCA (HEIJUNGS et al., 1992) made a major step forward by using a uniform toxicological basis, but still did not include the fate of a substance. Thus the potential toxic effects of readily degradable substances were still over-estimated by orders of magnitude in comparison to persistent substances like dioxins and metals. Aware of this drawback, the authors of the Dutch guide recently published an approach that includes fate information based on the Uniform System for Evaluating Substances (USES) (GUINÉE et al, 1996a). USES was originally developed as a support tool for substance policy. It had to give a first rapid, quantitative screening of the hazards and risks to man and the environment of a certain substance (VERMEIRE and V.D. ZANDT, 1995). USES consists basically of a Mackay Type-III fate model that calculates estimated concentrations of a substance in the various environmental compartments on the basis of substance properties like degradation rates in various media, vapour pressure and octanol-water partition coefficients. Based on these concentrations, the model is able to calculate the exposure of predators and humans making use of intake models.

3 Uncertainties in a Case on PVC

3.1 Introduction

TNO and CML applied the new USES-based approach in a substance flow analysis (SFA) on PVC in Sweden, commissioned by Norsk Hydro (TUKKER et al., 1996 and 1998). The flows of PVC in Sweden in 1994 were quantified, including the flows of PVC's precursors vinyl chloride, ethylene dichloride and chlorine. The emissions of all substances from the processes that take place in this material chain were inventoried, as far as they were causally related to the material PVC. All emissions from the production plant, for instance, were taken into account for the PVC production itself, including emissions of non-halogenated substances. Emissions by chain part were evaluated making use of LCIA. In order to assess the importance of the contribution of the PVC-chain to Swedish environmental problems in relation to PVC's importance to the Swedish economy, normalised LCIA-scores were also calculated for each chain part. For the related normalisation data, we refer to TUKKER and KLEIJN (1996). Thus, in fact, we created an approach that combined elements of SFA and LCA in order to give a truly overall insight in the impacts related to PVC on Swedish territory.

The main emissions from the system were several process emissions from the material production and waste management stage (e.g. VCM, EDC, dioxins, etc.), point source emissions from the manufacturing of PVC-products (mainly plasticisers used for flexible PVC, like phthalates) and diffuse emissions from the use stage (mainly evaporation and leaching of phthalates from flexible PVC). The most important phthalate included was DEHP, but in some applications other phthalates played a role (like BBP in flooring and DIDP in cable isolation). For an elaborated description of the method and data inventory we refer to TUKKER et al. (1996 and 1998). Here we will only discuss the results with regard to toxic releases.

3.2 Problems with USES-based equivalency factors for toxicity impacts

The new USES-based method for human toxicity gave a number of results that seemed not to be logical. For instance, we noted that the rather high phthalate emissions from coated fabrics lead to similar scores as the much lower emissions from soft tubes/hoses, in which a different plasticiser is used. Another problem was the score on terrestrial ecotoxicity: small emissions of DIDP from cables dominated the much higher emissions from other flexible PVC applications containing DEHP, like building plate. This was not expected since the toxicity and other substance properties of different phthalates are rather similar (KEMI, 1994; JANUS et al., 1994). Since all these results were directly a consequence of the values of the equivalency factors given in GUINÉE et al. (1996a), we decided to review the calculation of these equivalency factors. It appeared that the database with substance properties in USES contains outdated values, inappropriate default values or other errors. For instance, it appeared that for the calculation of the equivalency factor for vinyl chloride (VCM) USES had inserted a default value for photochemical degradation, since no specific value was given in the original database. This led to unacceptable high life time estimates: 160 days in air, where 1-2 days is generally assumed as a realistic value. Such errors resulted in high differences in equivalency factors for similar substances like DEHP and DIDP.

Therefore, a revised input data set for USES was created making use of state-of-the-art literature and available databases (ISIS, undated; IUCLID, 1996; VERSCHUEREN, 1996, DOSE, undated, HOWARD et al., 1991). This was done for the substances that had the highest contribution to toxicity scores: the phthalates DEHP, DIDP and BBP, VCM, ethylene dichloride (EDC), hexachlorobenzene and dioxins. For 4 of the 7 substances checked, a number of the original input parameters of USES had to be changed. On the basis of the comparison, a most probable average value for the input parameters was chosen, and a second-set equivalency factor was calculated (called the improved TNO/CML set).

This improvement lead to relatively high equivalency factors for phthalates. This result was questioned by the Euro-

pean trade organisation of phthalate producers, the European Council for Plasticisers and Intermediates (ECPI). ECPI is currently involved in preparing a hedset (harmonized electronic data set) for substance properties of a number of phthalates within the framework of the EU existing chemicals programme. They delivered us a third set of data for phthalates which was based on even more recent, partly unpublished work. For the solubility and the bioconcentration factors (BCFs) large differences were found between the existing databases and the ECPI data. Since these parameters are very important for the calculation of the equivalency factors, we decided to create a third set of equivalency factors. This ECPI data set was based on the work of STAPLES et al. (1996). They find that the differences between the solubility data for phthalates found in literature is mainly due to a number of experimental difficulties, like the inability to separate colloidal emulsions of undissolved chemicals in the aqueous phase, contamination from laboratory plastics and withdrawing samples through the surface film in which phthalates accumulate. All these problems can lead to experimental artefacts that yield measured values that overestimate the true water solubility (STAPLES et al., 1996). Furthermore, the group of STAPLES find that predictions of BCFs that rely on simple correlations with chemical hydrophobicity (as used in USES) are inappropriate if applied to phthalates since these models are based on data for persistent chemicals. LOWENBACH et al. (1995) suggested an alternative approach by extrapolating BCFs found with rodents to cattle (STAPLES et al., 1996). The BCFs found via this approach are a factor of 1000 to 10000 lower then the BCFs that were calculated by USES on the basis of hydrophobicity; as a consequence, with a dramatic change in values of the equivalency factors for phthalates.

3.3 A first, case-specific assessment of uncertainties

Some of the important differences in USES-input data we encountered in this process are reviewed in **Table 1**. **Table 2** reviews the effects on the values of the equivalency factors for toxicity themes for the 7 most important substances in our study. A more extended review than possible here is given in Annex III and IV of TUKKER et al. (1996) (→ *Appendix*, Top 1). The improved TNO/CML set results for VCM, DIDP, DEHP and BBP in quite different values as for the original LCA-USES set. The rather difficult to explain differences between the phthalates from the original LCA-USES set are diminished. Only BBP has a relatively high equivalency factor for emissions to water which is logical due to its relatively high water solubility compared to DEHP and DIDP. The differences in solubilities in the improved TNO/CML set and the ECPI set have little effect on the equivalency factors. The differences in BCFs are only important for human toxicity, and result in equivalency factors for the ECPI-set which are up to a factor of 500 to 1000 lower.

It goes without saying that such differences can have dramatic consequences for the results and policy conclusions of an LCA. For the 3 sets of equivalency factors based on the USES-model, **Figure 1** (*p. 250*) provides the normalised theme scores from all emissions from the Swedish PVC-chain together, and for 6 substances or substance groups emitted from the PVC-chain. For the sake of completeness, we further calculated normalised theme scores making use of equivalency factors of the CML guide of 1992 that did not yet include fate modelling (HEIJUNGS et al., 1992).

When the LCA-USES set is applied, the emissions of VCM dominates the contribution of the PVC-chain to human toxicity. But the toxicity score of the PVC-chain as such is very small compared to the Swedish total: only 0.002%, which is much lower than PVC's contribution to the Swedish Gross National Product (some 0.15%, see TUKKER et al., 1996a). With the improved TNO/CML set, the picture changes dramatically. VCM, so important with the LCA-USES set, is now invisible. Phthalates are now responsible for over 95% of PVC's score on human toxicity. Moreover, the normal-

Table 1: Some examples of differences in substance properties

Substance and data source*	Vapour pressure (Pa, 20-25 °C)	Kow	Photodegradation in air (DT_{50} in days)	Water solubility (mg/l)	BCF fish (kg/kg)**
DEHP					
LCA-USES (GUINÉE et al., 1996a)	8,6 E-04	5.24	1	0.045	8083
TNO/CML-set (literature average)	1 E-04	7.5	1	0.04	1.47 E+06
ECPI-set ***	1.3 E-5	7.5	1	0.003	120
IUCLID, 1996	1-8.6 E-04	4.8-9.6	0.6-2.2	0.007-0.04	114-886
ISIS (undated)	4.5 E-05	3-4	22	0.35	-
EDC					
LCA-USES (GUINÉE et al., 1996a)	24000	1.92	80.21	5500	3.87
TNO/CML-set (literature average)	8500	1.48	100	8700	1.41
IUCLID, 1996	8330-8700	1.45-1.46	12-120	8000-9000	-
VCM					
LCA-USES (GUINÉE et al., 1996a)	333000	0.6	160	1100	0.19
TNO/CML-set (literature average)	333000	1	2	1100	0.47
IUCLID, 1996	333000-340000	1.36-1.58	2.2-2.7	915-1100	<10

* For each data source, the value or range of values is given. We analysed more sources than given in this table. The ECPI set was irrelevant for EDC and VCM, since ECPI gave no values for those substances. See further in the main text
** For the LCA-USES-set and TNO/CML-set, all BCFs were calculated by USES on the basis of other input data
*** Based on STAPLES et al. (1996) and LOWENBACH et al. (1995)

Table 2: Toxicity equivalency factors from GUINÉE et al. (1996a; the LCA-USES set), corrected after comparison with databases for physical chemical properties (the improved TNO/CML-set) and factors based on physical chemical properties given by the ECPI (the improved ECPI-set)

Substance	CAS-no.	Equivalency factor	LCA-USES GUINÉE et al.	Improved TNO/CML-set	Improved ECPI-set	LCA-USES GUINÉE et al.	Improved TNO/CML-set	Improved ECPI-set	LCA-USES GUINÉE et al.	Improved TNO/CML-set	Improved ECPI-set
		for:	Air			Surface Water			Soil generic		
di(2-ethylhexyl)phthalate	117-81-7	aquatic ecosystem AETP	2.90E-01	3.91E-03	6.53E-03	4.70E+01	2.49E-01	2.49E-01	8.30E-04	2.99E-09	7.93E-09
		terrestrial ecosystem TETP	1.10E+02	2.07E+03	4.09E+03	2.60E+01	2.71E-00	9.05E-00	3.60E+03	1.30E+04	1.30E+04
		human HTP	1.40E+01	9.19E+04	6.45E+01	5.90E+01	2.84E+02	1.57E-01	2.90E-00	1.69E-00	1.46E-01
diisodecylphthalate	26761-40-0	aquatic ecosystem AETP	4.50E+01	1.69E-02	1.69E-02	1.50E+03	5.88E-01	5.88E-01	9.20E-02	6.42E-10	6.80E-09
		terrestrial ecosystem TETP	6.40E+05	4.59E+02	4.56E+02	5.30E+01	8.24E-04	3.81E-01	1.70E+06	1.30E+03	1.30E+03
		human HTP	4.60E+02	1.97E+05	6.86E+01	1.60E+02	2.22E+02	6.18E-02	1.50E+02	5.59E-00	1.51E-01
benzylbutylphthalate	85-68-7	aquatic ecosystem AETP	3.10E+01	7.71E-01	7.38E-01	4.80E+02	2.73E+01	2.73E+01	6.40E-01	2.50E-05	2.65E-05
		terrestrial ecosystem TETP	1.20E+05	2.85E+02	2.61E+02	7.60E+04	3.75E-02	6.69E-02	3.60E+05	8.54E+02	8.54E+02
		human HTP	4.30E+02	3.76E+02	3.43E+02	1.00E+02	4.26E-00	1.59E-01	8.30E+01	1.91E-01	1.91E-01
1,2-dichloroethane	107-06-2	aquatic ecosystem AETP	1.20E-02	6.73E-02	6.73E-02	5.70E-01	7.29E-01	7.29E-01	1.20E-02	5.52E-02	5.52E-02
		terrestrial ecosystem TETP	4.20E-00	1.14E+01	1.14E+01	4.20E-00	1.14E+01	1.14E+01	2.00E+02	4.15E+02	4.15E+02
		human HTP	6.90E+01	8.46E+01	8.46E+01	6.90E+01	8.68E+01	8.68E+01	6.80E+01	7.97E+01	7.97E+01
hexachlorobenzene	118-74-1	aquatic ecosystem AETP	4.30E-00	3.20E-03	3.20E-03	1.60E+02	1.27E+02	1.27E+02	3.40E-00	3.20E-03	3.20E-03
		terrestrial ecosystem TETP	4.00E+05	6.15E-00	6.15E-00	3.10E+05	5.00E-00	5.00E-00	2.40E+06	2.17E+01	2.17E+01
		human HTP	4.40E+03	2.77E+02	2.77E+02	7.30E+03	3.42E+03	3.42E+03	1.10E+04	2.85E+02	2.85E+02
2,3,7,8-TCDD	1746-01-6	aquatic ecosystem AETP	7.20E+06	7.23E+06	7.23E+06	1.50E+08	1.48E+08	1.48E+08	2.10E+06	2.06E+06	2.06E+06
		terrestrial ecosystem TETP	2.60E+08	2.62E+08	2.62E+08	2.90E+07	2.89E+07	2.89E+07	5.10E+08	5.10E+08	5.10E+08
		human HTP	2.60E+10	2.57E+10	2.57E+10	3.20E+09	3.15E+09	3.15E+09	4.70E+10	4.65E+10	4.65E+10
vinylchloride	75-01-4	aquatic ecosystem AETP	3.10E-03	3.88E-05	3.88E-05	4.60E-01	4.59E-01	4.59E-01	3.10E-03	3.85E-05	3.85E-05
		terrestrial ecosystem TETP	4.50E-02	6.46E-04	6.46E-04	4.40E-02	6.42E-04	6.42E-04	1.10E-00	1.26E-00	1.26E-00
		human HTP	5.50E+02	6.78E-00	6.78E-00	5.50E+02	1.11E+01	1.11E+01	5.40E+02	8.40E-00	8.40E-00

ised score on human toxicity from the PVC-chain is now only half of its contribution to the Swedish GNP and thus not negligible anymore. The policy conclusion would clearly be that phthalates emitted from the manufacturing of PVC-products and the use stage are the key issue with regard to human toxicity in the Swedish PVC-chain. But use of a third set, the ECPI set, once again would turn the tables. The contribution of the PVC-chain to the Swedish total diminishes dramatically and phthalates are hardly visible anymore. The new priorities are dioxins and to some extent EDC, calling for emission reductions in the EDC/VCM production and waste incineration. Finally, if we would have applied the old CML manual from 1992, once again other priorities would have arisen. EDC and VCM now are the most important emissions. NO_x emitted from a EDC-cracking furnace, no issue in any of the other analyses, all of a sudden becomes important. The total score of the PVC-chain, with 0.036%, is all of a sudden a tangible fraction of the Swedish total score.

In this comparison, particularly the dramatic difference between the improved TNO/CML set and the ECPI set is reason for concern. Despite their contradictory outcomes they are based on a state-of-the-art fate and exposure model, and two equally acceptable data sets for substance properties (→ *Appendix*, Top 2).

Several taxonomies of uncertainty have been developed. Most authors include two types of uncertainties: uncertainties that relate to quantitative data and the appropriateness of models in describing the real world, and uncertainties related to implicit choices in problem definitions and value judgements in a method (WYNNE, 1993; DOUGLAS and WILDAVSKY, 1982). For a review we refer to V.D. SLUIJS and SCHULTE-FISHEDICK (1997). For the purpose of this paper, we will apply a taxonomy developed by ROTMANS et al. (1994):

1. data uncertainties: uncertainties in input data in a model;
2. modelling uncertainties: uncertainties with regard to the model structure, the equations and parameter values;
3. paradigm uncertainties: uncertainties due to the fact that a problem may be defined and analysed from different scientific perspectives.

4.2 An analysis of uncertainty by type

The *data uncertainty* aspect concerns the input data like tolerable daily intakes (TDIs), BCFs, vapour pressures, etc. used to calculate equivalency factors with USES. Mainly this type of uncertainty, limited to substance-related fate data, has

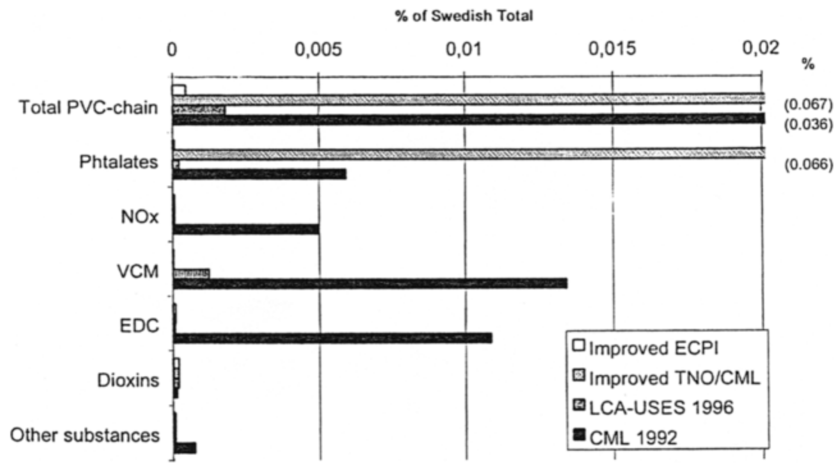

Fig. 1: Contribution of substance groups emitted from teh Swedish PVC-chain to the total Swedish score on human toxicity for 4 sets of equivalency factors

4 Towards a Structural Analysis of Impact Assessment of Toxic Releases

4.1 Introduction

The case in the former section does not yet indicate if the encountered major uncertainties in impact assessment of toxic releases have a structural character. To analyse this point, we will make a distinction between levels of uncertainty and indicate what determines the magnitude of each level of uncertainty.

been made visible in the case. **Table 1** illustrates that the following types of problems exist:

1. in some cases, like EDC here, just wrong basic data for substance properties might have been used;
2. in some cases, like VCM and the phthalates here, inappropriate default values for substance properties have been used;
3. in other cases, literature is not unanimous about the value of key substance properties that form an input in USES, or data is simply lacking;

4. finally, for some substances it is questionable if the current literature values of substance properties have been assessed with an appropriate method (like here solubility and BCFs for phthalates).

Of course some of these problems have no structural character and can relatively easily be tackled, like point 1. But toxicological literature shows that the magnitude of all these data gaps and uncertainties can hardly be overestimated. In society about 100,000 chemicals are in use, of which 2,000 are used in quantities over 1,000 ton a year in the EU. It is estimated that even for these 2,000 High Production Volume Chemicals (HPVCs) a minimum screening information data set is available that allows for a first, rough insight in substance properties and toxic effects in only 5 to 10% of the cases (VAN LEEUWEN, 1995). Recent, thorough analyses of the situation in the US confirmed the extent of these data gaps (CRANOR, 1993, p. 27; EDF, 1997; US EPA, 1998). For the substances for which data is available, it is assumed that the assessment protocols leave room for outcomes that differ one order of magnitude or more for the toxicological limit values alone, for example due to the choice of the high-dose to low-dose extrapolation procedure, differences in interspecies scaling factors, etc. (CRANOR, 1993, p. 22; MCCRAY, 1983). Recently, the book "Our Stolen Future" popularised another warning: it can even not be assumed that toxicological risk assessment has discovered all relevant types of toxic effects as such, as illustrated by the emerging debate on endocrine disrupters (COLBORN et al., 1996).

The *modelling uncertainty* aspect concerns uncertainties that are introduced by the structure of USES itself. USES does not take into account the fate and effects of a substance's degradation products, nor effects due to exposure to a combination of substances (GUINÉE et al., 1996b). Further, the fate and intake modelling in USES is a simplification of reality, and may even not include relevant exposure routes. HUNT (1994) mentions the example that certain fat soluble contaminants may concentrate in the surface microlayer, a thin film of natural oils in surface water and sea. Eggs of certain fish species that float close to the surface may come in contact with this layer. This may lead to severe reproductive effects, even at very low predicted concentrations of contaminants in water which were never regarded as a problem. In the case discussed in section 3, the influence of modelling uncertainty to some extent has been made visible by comparing results with toxicity impact assessment methods that include and exclude fate modelling.

The *paradigmatic uncertainty* aspect concerns the fact that different scientific perspectives are possible to analyse a problem. These have not been made visible in our case, but can be illustrated as follows: LCIA of toxic releases is simply based on the assumption that knowledge is adequate to perform a meaningful fate and effect modelling. Moreover, the residence time of a toxic substance in the environment is not treated as a parameter of special interest. What matters are predicted potential environmental concentrations and intakes – whether these are the result of large emissions of a substance with a high degradation rate or smaller emissions of a persistent substance is irrelevant for the method. However, for the manageability of the situation and the possibility to correct wrong judgements, the difference in residence time (or persistence) is crucial. Groups like Greenpeace simply disagree with the basic premise of LCIA: they fear that data and modelling uncertainties will be so high that major errors in toxicity assessments will frequently occur (COLBORN et al., 1996; IJC, 1993). If this concerns a degradable substance, mistakes can be corrected on short notice by diminishing emissions, but this does not work for persistent substances: emitted quantities from the past will be present for years, or decades to come in the environment. These aspects, that are two of the main reasons for conflicts between industry and environmentalists on toxic substances, are simply disregarded by the current methods for LCIA of toxic releases (TUKKER, forthcoming). In this respect, it is useful to note that forums like the International Joint Commission for the Great Lakes in Canada and the US (IJC) have explicitly decided to base their substance policy on the assumption that fate and effect modelling is an inappropriate basis for dealing with the risks of toxic substances. The IJC regards particularly the risks related to persistent substances as unassessable and thus unmanageable. Such substances therefore should be eliminated (IJC, 1993). A similar philosophy has been proposed by the Swedish National Chemicals Inspectorate (KEMI, 1991) and more recently by the Swedish Chemicals Policy Committee (→ *Appendix*, Top 3).

4.3 Are there easy solutions?

It is generally recognised that the research needed to deal with the data and modelling uncertainties alone would cost decades, if not centuries, and is virtually unaffordable (IJC, 1993). Under the EU, existing chemicals programme information comes available for only about 20 to 50 substances a year. This means that 40 to 100 years is required even to gather information for the 2000 HPVCs. And even then such fundamental problems like combination toxicity, the translation of animal test data to safe levels in other species, high-dose to low-dose extrapolation, and the possibility that yet unknown effects will be discovered, are left unaddressed.

Given the tremendous data gaps, there is therefore no choice in toxicological risk assessment but to apply stringent priority setting procedures, supported by step-wise approaches to assess risks. In case of low data availability with regard to substance properties, conservative estimates in all extrapolation and estimation procedures for substance properties are initially used. With such data, a first rapid, quantitative screening of the hazards and risks to man and the environment is performed with models like USES. If a risk is predicted, additional research is done to get a better view on substance properties (VERMEIRE and V.D. ZANDT, 1995). But in other cases there will be little incentive to perform additional tests. In each improvement cycle, the main aim is to

prevent a false negative outcome in the evaluation (i.e. to judge that there is no risk, in a situation where there is a risk). In brief, this will lead to a situation where a rather good insight in substance properties will exist for a relatively small number of substances; for a large other set there will be mainly conservative estimates, and the majority of substances used in society will hardly demonstrate any data at all for the time being. It has to be stressed that this problem is much more important in LCA than for the application for which USES was designed. USES was developed as a quick screening tool for a preliminary risk assessment, aimed at giving conservative answers on the question of whether a certain risk threshold may be exceeded. This is a decision with a simple yes-no character. LCIA includes the task of comparing the potential effects of substances: a relative, quantitative comparison. For this purpose, it is mandatory that in the average case, equal quality data be available for all relevant substances – a goal irrelevant for substance policy and, in fact, a situation that substance policy, due to the enormous costs, deliberately avoids by step-wise approaches and priority setting schemes, in which USES plays a key role. In sum: models like USES may contribute to solving problems in the field of the risk assessment of chemicals. But this does not automatically mean that such a model can be directly applied to solve problems in a field that has a different context, such as LCA (→ *Appendix,* Top 4).

5 The Practice of Dealing with Uncertainty

5.1 The field of LCA

Until now, the LCA community has dealt only with partial success with uncertainty and ignorance. It is generally recognised that the development of methods to deal with uncertainty in LCA is one of the research priorities for the coming years (FINNVEDEN and LINDFORS, 1997; UDO DE HAES, 1996). Most LCA manuals explicitly warn for the shortcomings of certain approaches, specifically in the case of toxicity impact assessment (HEIJUNGS et al., 1992; GUINÉE et al., 1996b). Several LCAs have been performed in which results concerning toxicity were very prudently shown (FINNVEDEN et al., 1996), or even deliberately excluded from the scope (GÜNTHER and LANGOWSKI, 1997). At the same time, it seems that uncertainties, in practice, are neglected or underestimated on a large scale. Respected manuals like the 1992 CML guide, the recent report on the USES method, and the related software tools, due to the form in which they have been made available, are strong invitations to apply toxicity impact assessment methods that probably produce close to invalid results in quite a number of cases, irrespective of the warnings these documents may contain. They are, after all, written as manuals: booklets that are easily perceived by a scientific community as exemplars guiding how to perform state-of-the-art research in the field at stake (compare KUHN, 1962). In the period between 1992 and 1996, in numerous LCAs toxicity scores must have been produced, in most cases without questioning them despite their totally lacking fate modelling. We restrict our discussion to some personal examples. In the Dutch chlorine chain study, TNO and CML based their conclusions with regard to the toxicity risks of chlorine mainly on risk assessment, but produced as a first approach also normalised toxicity theme scores using the 1992 CML Guide (TUKKER et al., 1995; KLEIJN et al., 1997). They were widely communicated by industry (e.g. NCI, 1995), which lead to strong and negative comments on the LCA method (KOEMAN et al., 1995; COPIUS PEEREBOOM, 1996). The problems reported in section 3 with the USES-based method were fully unforeseen (→ *Appendix,* Top 5). CML felt it was necessary to send an explicit warning to those who had ordered the report that described the USES-method that was published months before (CML, 1996). Still, this warning only indicated the problem of default values in USES, and not the other problems indicated in section 3.2 in this paper. The recent strategic research programme of LCANET mainly addresses data uncertainty and to some extent model uncertainty, but does not mention paradigmatic uncertainty (UDO DE HAES and WRISBERG, 1997). Several documents discussing research needs in LCIA plea for the development of more structural approaches to link inventory data or characterisation data with safeguard subjects – while such links may be surrounded with even more ignorance than the USES model in toxicity impact assessment. Underestimating uncertainty thus seems not to be an individual, but at least a partially collective process within the LCA-community (→ *Appendix,* Top 5a).

5.2 Other areas

A bit to our defence, the LCA-community is not unique in underestimating uncertainty. But at the same time this is a warning of how difficult it is to avoid this. The branch of the scientific community that studies how sciences develop themselves and make progress, the so-called sociology of science, has shown such struggles in dealing with uncertainty are more routine than an exception. The numerous case studies published in Journals like Social Studies of Science and Science and Public Policy show that what scientists often present as independent knowledge is at best a mixture of facts and constructs: unconscious conventions within the scientific community at stake, implicit assumptions, and hidden value judgements. This is not necessarily negative: the existence of such paradigms prevents scientists from having to question every piece of basic knowledge they use, and allows them to concentrate on the specific problem they want to solve (KUHN, 1962). Such processes have been described in fields as diverse as energy policy (THOMPSON, 1984), geology (BOWDEN, 1985), climate change (SHACKLEY and WYNNE, 1995; V.D. SLUIJS, 1997), health care (KNORR-CETINA, 1985) and solar neutrino science (PINCH, 1981). This leads to the situation that facts are not necessarily accepted as authoritative because they reflect some truth, but because they are validated through processes of informal negotiation and can be ranged into frameworks of shared assumptions and interferences (JASANOFF, 1987; 1990).

Authors like WEINBERG (1972) have sought the answer in a rigorous separation of science from what he called trans-science: issues that "hang on the answers to questions which can be asked of science and yet which cannot be answered by science". Trans-scientific questions should in his view mainly be solved in a political setting. But in practice it is often observed that the opposite occurs (JASANOFF, 1987; WYNNE, 1987). RIP (1992) noted that it is highly questionable if the current risk assessment systems for external safety can predict actual or potential risks with a meaningful certainty. But it has the important advantage that the calculations are standardised, which makes the decision making process much more predictable – and the system attractive – for both industry and the authorities. In WYNNE's observation on areas like external risk assessments of nuclear power plants, for example, regulatory bureaucracies are engaged in the conversion of sheer ignorance and implicit conflicts of perspective into manageable uncertainty (WYNNE, 1987, p. 312). Driving forces behind such processes are, among other things, the demand of decision makers and the public for at least the appearance of certainty (FUNTOWICZ and RAVETZ, 1990; SMITHSON, 1993), the mainstream approach to risk assessment in the eighties (where relatively well-structured, standardised engineering systems like chemical plants dominated as cases, and this approach seemed to easily adapt to the situation of nuclear plants) and the political climate of demand for the public justification of policy. By such mutual construction processes, practical knowledge is produced. An appearance of factual solidity of science is created by legitimate, institutional agreements. WYNNE (1987; 1993) refuses to call the choices made in such processes value choices that can be made explicit during scientific research. This does not take into account their depth and subtlety: individual scientists are just not aware of them, let alone they cannot identify and present them. On the contrary: if ignorance is revealed, the first reaction is in most cases to shoot the messenger rather than developing strategies for managing under ignorance (SMITHSON, 1993).

5.3 Consequences for LCA impact assessment of toxic releases

The consequences are obvious. Such practical, so-called constructed knowledge will survive as long as societal actors have no reason or not enough power to start a conflict. But if, like in the case of the Dutch chlorine study, there is strong lack of consensus, (constructed) knowledge is put to test and thus subject to deconstruction, which leads to a good chance that until then hidden uncertainties or choices are made explicit (RIP, 1986). The credibility of decision making, predictably, is negatively affected (WYNNE, 1987, p. 351). Since ignorance has been converted into apparently manageable uncertainty, another negative consequence is that this science cannot easily develop strategies to cope with surprise, and thus might enhance factual risk (WYNNE, 1987, p. 325) (→ *Appendix*, Top 6).

In the specific case of Life Cycle Impact Assessment of toxic releases, results should at least have enough validity to support one of LCA's generally recognised application areas, such as (V.D. BERG et al., 1995):

1. product comparisons;
2. to help the design of new products or services;
3. to indicate strategically the direction of development;
4. monitoring the environmental impacts of a product, or comparing it with a benchmark;
5. indicating the environmentally dominant stages in a product life cycle.

The case and the subsequent analysis seem to leave little room but for the conclusion that in toxicity impact assessment uncertainty and ignorance are the dominant factors, and that even sophisticated methods like the USES-approach at this moment can do little more than to produce questionable results. An impact assessment method with such inherent weaknesses can serve little of the application areas mentioned. Defining the goals and limits of LCA in terms of a Less is better concept (WHITE et al., 1995) does not much change the situation. The key point is that, depending on the data set, the fate and exposure model, and the evaluative paradigm applied, different substances will dominate the others. Using one set of assumptions, less emissions of a certain substance will hardly seem relevant, where using other sets, less emissions of the same substance will be regarded as the key factor for environmental success. Even a fruitful use in the less demanding applications like 4 and 5 will thus be frustrated, let alone in application 1 to 3, that have the additional complication that in general comparisons between rather different systems are necessary.

6 An Outlook for a Solution and Conclusions

Altogether, this paper results in a number of worrisome and hard questions about the current practices of dealing with toxicity in LCA. The case in section 3 illustrated that uncertainty in state-of-the-art fate input data alone can result in fully contradictory policy advice with regard to toxicity problems. Section 4 made plausible that such uncertainties are most probably structural, and suggested strongly that equally important uncertainties may be at stake for effect data and fate models as such. Finally, section 4 indicated that, given these large uncertainties, different groups in society develop different evaluative paradigms to judge a situation. The relevant types of uncertainty are visualised in **Table 3**, specified by step in the emission effect-chain.

We must reluctantly conclude that these levels of uncertainty have been underaddressed in the practice of LCIA to date. Even methods that lacked essential elements such as fate

Table 3: Types of uncertainty with regard to emissions, fate and effects of toxic substances

Type of uncertainty	Step: Emission	Step: Fate	Step: Effect
Data uncertainty	Type and amounts of substances emitted	Substance properties (fate) Environmental information	Substance properties (toxicity)
Modelling uncertainty	n.r	Extent to which the model reflects the real world	n.r.
Paradigmatic uncertainty	Tacit biases and assumptions	Tacit biases and assumptions	Tacit biases and assumptions

n.r: not relevant
grey: type of uncertainty illustrated in the case

modelling have been used for a considerable period as state-of-the-art tools to produce policy advice. LCIA of toxic releases thus seems to fall into the trap so many areas of science for policy have fallen into. In the (tempting) quest for a single calculation rule, the interpretative space is narrowed by a negotiation process within a specific community – in part probably unconsciously and/or informally. Ignorance is converted into an apparent manageable uncertainty, and plausible, thoughtful alternative evaluative perspectives of other important actor coalitions in society are simply excluded or overruled.

How can this situation be solved? Of course, one could refrain from impact assessment. One could try to come to a judgement on the basis of rather robust emission inventory data and rudimentary logic – or otherwise simply acknowledge that one does not know (→ *Appendix*, Top 7). However, this is not an ideal solution. There is the danger that a judgement is still made, but only implicitly (JOLLIET, 1996).

In our view, the key to a solution is that the LCA community truly accepts the fact that LCIA of toxic releases is a far cry from the type of science that gave us robust calculation rules such as Ohm's law. As a consequence, we should get rid of the idea that we can assess such a thing as a single best truth in setting toxicity priorities in LCA. Policy sciences have convincingly shown that in the case of such important uncertainties, and a strong lack of consensus (such as in the PVC-debate). In general, a few evaluative paradigms play a role, which are equally acceptable, also in scientific terms. The starting point should therefore be an analysis of these different evaluative paradigms (see for toxicity: TUKKER, forthcoming). The most plausible and consistent ones can be converted into LCIA indicator systems that reflect the different paradigmatic perspectives of stakeholders (→ *Appendix*, Top 8). The best example currently available is the indicator system developed by ROTMANS and DE VRIES (1997), who made use of the scientific perspectives that can be derived from the cultural theory of THOMPSON, SCHWARZ, DOUGLAS, and others (THOMPSON et al., 1990) (→ *Appendix*, Top 9). Once the paradigmatic uncertainties have been dealt with along such approaches, the more classical modelling and data uncertainty can be addressed (HEIJUNGS, 1996). Only by this strategy does the LCA community live up to Weinberg's rules. The policymakers are supplied with information that do justice to the existence of different, but equally plausible, consistent and thus acceptable types of logic. The final decision on the trans-scientific aspects is left to the democratic decision making process, instead of to the less democratic decision making process based on the notion of an (apparent, but non-existing) scientific truth (→ *Appendix*, Top 10). Note this solution may imply that the current mainstream in LCIA development, that heavily relies on searching for one consensus method preferably based on models that link environmental interventions with safeguard subjects, has to be rejected as a starting point (→ *Appendix*, Top 11).

In our view, until approaches have been worked out that deal with paradigmatic, modelling and data uncertainty, all current methods for toxicity impact assessment should be very carefully applied. Practitioners that simply state that LCIA is unable to draw conclusions about toxic effects for the moment (GÜNTHER and LANGOWSKI, 1997) probably do a better job than those performing an impact assessment and presenting results unconscious of their weaknesses. In this respect, there may be an even more important message for those involved in the development of manuals for LCIA. Such manuals have the function of exemplars, and are invitations to apply the methods presented in the manual without too many thoughts about the strengths and weaknesses of the method. It is therefore strongly suggested to refrain from publishing anything that looks like a guide for impact assessment, as far as it concerns themes that probably suffer from high uncertainties and as long as the guide is not able to include uncertainty in the impact assessment method (→ *Appendix*, Top 12).

7 References

AMSTERDAMSKA, O. and R. HAGENDIJK (1990): De nieuwe zekerheden van het hedendaagse wetenschapsonderzoek [The new certainties of present day's research of science]. Kennis en methode 1990/1

BERG, N.V.D. et al. (1995): Beginning LCA. Unilever and CML, Leiden, Holland

BOWDEN, G. (1985): The Social Construction of Validity in Estimates of US Crude Oil Reserves. Social Studies of Science, Vol. 15, 207-40

BRAUNSCHWEIG, A., R. FORSTER, P. HOFSTETTER, R. MUELLER-WENCK (1996): Developments in LCA valuation. IWO Diskussionspapier Nr. 32, St. Gallen, Switzerland

CML (1996): Letter concerning default data in USES to the readers of Guinée et al. (1996a). CML, Leiden, 1996

CONSOLI, F., D. ALLEN, I. BOUSTEAD, J. FAVA, W. FRANKLIN, A.A. JENSEN, N. DE OUDE, R. PARRISH, R. PERRIMAN, D. POSTLEWAITE, B. QUAY, J. SEGUIN and B. VIGON (1993): Guidelines for Life-Cycle Assessment: A Code of Practice. SETAC, Brussels, Belgium

COLBORN, T., D. DUMANOSKI and J.P. MAYERS (1996): Our stolen future: are we threatening our fertility, intelligence and survival. Penguin Books Inc. USA

COPIUS PEEREBOOM, J.W. (1996): A disputable chlorine balance. (in Dutch), Milieu 1996/5, Boom Publishers, Meppel, The Netherlands

COWAN, C.E., D. MACKAY, T.C.J. FEITEL, D. V.D. MEENT, A. DI GUARDO, J. DAVIES and N. MACKAY (1995): The Multimedia Fate Model: A vital tool for prediction of the fate of chemicals. SETAC Press, Pensacola, Florida, USA

CRANOR, C.F. (1993): Regulating toxic substances. A philosophy of science and the law. Oxford university press, New York, USA

DOSE (undated, continuous updates), The Dictionary of Substances and their effects. CD-ROM, The Royal Society of Chemistry, Cambridge, UK

DOUGLAS, M. and A. WILDAVSKY (1982): Risk and Culture. University of California Press, Berkeley, USA

EDF (Environmental Defense Fund) (1997): Toxic Ignorance. EDF, Washington D.C., USA (also available at: www.edf.org)

FUNTOWICZ, S.O. and J.R. RAVETZ (1990): Uncertainty and quality in science for policy. Theory and Decision Library, ser. A, Vol. 15, Kluwer, Dordrecht, Holland

FINNVEDEN, G. L-G. LINDFORS, and L. WENNBERG (1996): Alternativa material till PVC. IVL Rapport B 1232, IVL, Stockholm, Sweden

FINNVEDEN, G. and L-G. LINDFORS (1997): Life cycle impact assessment and interpretation. In: UDO DE HAES and WRISBERG (1997)

FINNVEDEN, G. (1997): Letter to the editor. Int. J. LCA 2 (4) 185

GÜNTHER, A. and H-C. LANGOWSKI (1997): Life Cycle Assessment Study on Resilient Floor Coverings. Int. J. LCA 2 (2) 73-80

GUINÉE, J. B., R. HEIJUNGS, L. VAN OERS, D. V.D. MEENT, T. VERMEIRE and M. RIKKEN (1996a): LCA Impact assessment of toxic releases. CML/RIVM, Product policy series 1996/21, Ministry for the Environment, The Hague, Holland

GUINÉE, J.B. R. HEIJUNGS, L. VAN OERS, D. V.D. MEENT, T. VERMEIRE and M. RIKKEN (1996b): USES. Uniform System for the Evaluation of Substances. Inclusion of Fate in LCA Characterization of Toxic Releases Applying USES 1.0. Int. J. LCA 1 (3) 133-138

HEIJUNGS, R., J.B. GUINÉE, G. HUPPES, R.M. LANKREIJER, H.A. UDO DE HAES, A. WEGENER SLEESWIJK, A.A.M. ANSEMS, P.G. EGGELS, R. VAN DUIN and H.P. DE GOEDE (1992): Environmental Lifecycle Assessment of Products, Guide and backgrounds. CML/TNO/B&G, Leiden, Holland

HEIJUNGS, R. (1996): Identification of key issues for further investigation in improving the reliability of life-cycle assessments. J. Cleaner Prod. 4 (3/4) 159-166

HOFSTETTER, P. (forthcoming): Perspectives in Life Cycle Impact Assessment: A structured approach to combine models of the technosphere, ecosphere and valuesphere (Provisional title). Ph.D. thesis, ETH Zürich, Switzerland, expected publication date: autumn 1998

HUNT, J. (1994): The social construction of precaution. In: O' RIORDAN, T. and J. CAMERON: Interpreting the precautionary principle. Earthscan, London, UK

IJC (International Joint Commission) (1993): A strategy for virtual elimination of persistent toxic substances. Volume 1 and 2, Report of the Virtual Elimination Task Force to the International Joint Commission, Windsor, Ontario, Canada

ISIS (undated), Internationaal Stoffen Systeem. Versie 5.1. HASKONING, Nijmegen, Holland

IUCLID (1996): International Uniform Chemical Information Database, Existing Chemicals. European Chemical Bureau, European Commission-JRC, Environment Institute,Ispra (Italy)

JASANOFF, S.S. (1987): Contested Boundaries in Policy-Relevant Science. Social Studies of Science, Vol. 17, 195-230

JASANOFF, S.S. (1990): The fifth branch: Science Advisors as Policymakers. Harvard University Press, Cambridge, MA, USA

JOLLIET, O. (1996): Impact assessment of human and eco-toxicity in life cycle assessment. In: UDO DE HAES (1996)

JANUS, J.A., J.M. HESSE, M.G.J. RIKKEN (eds.) (1994): Aandachtstoffen in het Nederlandse Milieubeleid – overzicht 1994 [Attention Substances in the Dutch Environmental Policy – overview 1994]. RIVM-report no. 601014006. Bilthoven, The Netherlands

KEMI (1991): Risk reduction of chemicals. Report 1991/1, KemI, Solna, Sweden

KEMI (1994): Phthalic Acid Esters Used as Plastic Additives. Report 1994/12 KemI, Solna, Sweden

KLEIJN, R., A. TUKKER and E. V.D. VOET (1997): Chlorine in the Netherlands, part I: An Overview. Journal for Industrial Ecology nr. 1: 1

KLÖPFFER, W. (1996): Reductionism versus Expansionism in LCA. Int. J. LCA 1 (2) 61

KOEMAN, J.H., H.A.J. GOVERS, W.A. HAFKAMP, A. VAN DER MEIDEN and H.J.W. SAS (1995): Peer review "Closing the chlorine chain". Apeldoorn, 17 and 18 August 1995, In: TUKKER et al. (1995)

KNORR-CETINA, K. (1985): Germ Warfare, review of: B.LATOUR, Les Microbes: guerre et paix suivi de Irreductions. Social Studies of Science, Vol. 15, 577-85

KUHN, T.S. (1962/1970): The structure of scientific revolutions. Dutch translation, Boom, Meppel, Holland

LATOUR, B. (1987): Science in action. Dutch translation 1995, Ooievaar Pockethouse, Amsterdam, Holland

LEEUWEN, C.J. VAN (1995): General introduction. In: LEEUWEN, C.J. VAN and J.L.M. HERMENS (eds.): Risk Assessment of Chemicals: An Introduction. Kluwer Academic Publishers, Dordrecht/ Boston/London

LINDFORS, L-G. (eds.) (1995): LCA-Nordic. Reports 1-9 and 10, TemaNord 1995:20, Nordic council of ministers, Copenhagen, Denmark

LOWENBACH, W.A., R. GOLDEN, E. REESE (1995): A model for estimating Beef and Milk Bioconcentration Factors for DEHP. Environmental Risk Sciences, Washington DC

MCCRAY, L.E. (1983): An anatomy of risk assessment: scientific and extra scientific components in the assessment of scientific data on cancer risks. In National Academy of Sciences: Risk Assessment in the Federal Government: Managing the process. Working papers, National Academy press, Washington DC, USA

NCI (1995): Chloor op zijn plaats [Chlorine in the position it deserves] and Risico's van chloor beheersbaar [Risks of chlorine manageable]. Journal of the Dutch Chemical Industry, 25 November 1995, Leidschendam, Holland

OWENS, J.W. (1996): LCA Impact Assessment Categories. Technical Feasibility and Accuracy. Int. J. LCA 1 (3) 151-158

PINCH, T.J. (1981): The Sun-Set: The presentation of Certainty in Scientific Life. Social Studies of Science, Vol. 11, 131-58

RIP, A. (1986): Controversies as Informal Technology Assessment. Knowledge Vol. 8 No. 2, pp. 349-371

RIP, A. (1992): Risicocontroverses en de verwevenheid van wetenschap en politiek [Controversies about Risks and the Links between Science and Politics]. Kennis en Methode 1992/1

ROTMANS, J., M.B.A. VAN ASSELT, A.J. DE BRUIN, M.G.J. DEN ELZEN, J. DE GREEF, H. HILDERINK, A.Y. HOEKSTRA, M.A. JANSSEN, H.W. KOSTER, W.J.M. MARTENS, L.W. NIESSEN, H.J.M. DE VRIES (1994): Global change and sustainable development, a modelling perspective for the next decade. Report 461502004, RIVM, Bilthoven, The Netherlands

ROTMANS, J. and H.J.M. DE VRIES (eds.) (1997): Perspectives on global futures: The TARGETS approach. Cambridge University Press, Cambridge

SHACKLEY, S. and B. WYNNE (1995): Mutual Construction - Global climate change: the mutual construction of an emergent science-policy domain. Science and Public Policy, Vol. 22, No. 4, pp. 218-230

SLUIJS, J. V.D. (1997): Anchoring amid uncertainty. On the management of uncertainties in risk assessment of anthropogenic climate change. Ph.D. thesis, University of Utrecht, Utrecht, the Netherlands

SLUIJS, J. V.D. and K. SCHULTE-FISHEDICK (1997): Omgaan met onzekerheden in wetenschap voor (milieu)beleid. Een inventarisatie van theorieën en aanpakken [Dealing with uncertainty in (environmental) policy. An inventory of theories and approaches. Rathenau Instituut, The Hague, The Netherlands

SMITHSON, M. (1993): Ignorance and Science. Knowledge Vol. 15 No. 2, 133-156

STAPLES C.A., D.R. PETERSON, T.F. PARKERTON and W.J. ADAMS (1996): A literature review: The environmental fate of phthalate esters. Submitted to Chemosphere

THOMPSON, M. (1984): Among the Energy Tribes: A Cultural Framework for the Analysis and Design for Energy Policy. Policy Sciences 17, pp. 321-339

THOMPSON, M., R. ELLIS and A. WILDAVSKY (1990): Cultural theory. Westview press, Boulder, CO, USA

TUKKER, A., R. KLEIJN, E.V.D. VOET, M. ALKEMADE, J. BROUWER, H. DE GROOT, J. DE KONING, T. PULLES, E. SMEETS and J.J.D. VAN DER STEEN (1995): A chlorine balance for the Netherlands. Report 95/40, TNO-STB and CML, Apeldoorn, Holland

TUKKER, A., R. KLEIJN, L. VAN OERS and E. SMEETS (1996): A PVC substance flow analysis for Sweden. TNO and CML, report 96/48, Apeldoorn, Holland

TUKKER, A. and R. KLEIJN (1996): Tentative LCA-normalisation data for Sweden. TNO/CML, report 96/49, Apeldoorn, Holland

TUKKER, A., R. KLEIJN, E V.D. VOET and E. SMEETS (1997): Chlorine in the Netherlands Part II: Risk management in uncertainty. Journal of Industrial Ecology Vol. 1, No. 2

TUKKER, A., R. KLEIJN, L. VAN OERS and E. SMEETS (1998): Combining SFA and LCA: the Swedish PVC analysis. Journal of Industrial Ecology Vol. 1., no 4

TUKKER, A. (forthcoming): Frames in the Toxicity Controversy. Kluwer Academic Publishers, Dordrecht/Boston/London. Expeced publication date: Autumn 1998. Also published as Ph.D. Thesis, Tilburg University/TNO, The Netherlands

VERMEIRE, T.G., M.E. VAN APELDOORN, J.C. DE FOUW, P.J.C.M. JANSEN (1991): Voorstel voor de humaan-toxicologische onderbouwing van C-(toetsings)waarden [Proposal for the human toxicological underpinning of C (testing) values]. RIVM report no. 725201005, Bilthoven

VERMEIRE, T. and P. VAN DER ZANDT (1995): Procedures of hazard and risk assessment. In: LEEUWEN, C.J. VAN and J.L.M. HERMENS (eds.): Risk Assessment of Chemicals: An Introduction. Kluwer Academic Publishers, Dordrecht/Boston/London, pp. 293-337

VERSCHUEREN K. (1996), Handbook of Environmental Data on Organic Chemicals. Van Nostrand Reinhold, USA

UDO DE HAES, H.A. (ed.) (1996): Towards a methodology for Life Cycle Impact Assessment. SETAC-Europe, Brussels, Belgium

UDO DE HAES, H.A. and N. WRISBERG (1997): LCANET - A Strategic Research Programme for Life Cycle Impact Assessment. LCA Documents, Ecomed Publishers, Germany

US EPA (1998): Chemical Hazard Data Availability Study. EPA's 1998 Baseline of Hazard Information that is Readily Avialable to the Public. EPA's Office for Pollution Prevention and Toxics, Washington, USA (available at: www.epa.gov/opptintr/ chemtest/hazchem.htm)

WEINBERG, A.M. (1972): Science and Trans-science. Minerva Vol. X Nr. 2, pp. 209-222

WHITE, P., B. DE SMET, H.A. UDO DE HAES and R. HEIJUNGS (1995): LCA Back on Track, But is it one track or two? LCA-News (SETAC Europe) 5 No 3 p. 2-4

WYNNE, B. (ed.) (1987): Risk management and hazardous waste. Implementation and the Dialectics of Credibility. Springer-Verlag, Berlin/London

WYNNE, B. (1993): Uncertainty and environmental learning. In: T. JACKSON (ed.): Clean Production Strategies. Lewis Publishers, Boca Raton/Ann Arbor/London/Tokio

Appendix

1. Part III of TUKKER et al. (1996) provides a full review of USES-input data we applied to calculate the different equivalency factors. Part IV of that study gives spreadsheets with the theme scores on human toxicity for the improved TNO/CML-set and the ECPI-set, including the contribution of each substance emitted from the Swedish PVC chain. Simple addition of the scores/ percentages for each individual emission of a substance (group) to water, soil and air provides the percentages in **Figure 1**. Part IV of TUKKER et al. (1996) also provides the emissions in kg for each individual substance. Furthermore, the equivalency factors relevant to the LCA-USES set can be found in **Table 2**, as well as the equivalency factors relevant for the CML 1992 set in HEIJUNGS et al. (1992). With this data, the calculation of the percentages in the figure is straightforward. All normalisation data relevant for the percentages in the notes of the table can be found in TUKKER and KLEIJN (1996). In TUKKER et al (1996) it was not useful at the end to include all figures or calculations made earlier with the erroneous equivalency factors of the LCA-USES set, let alone calculations based on the 1992 CML guide. The detailed spreadsheets of the latter calculations are therefore not included in TUKKER et al. (1996). They are also too long to be included in this paper. However, they are available on request from the author.

2. In the same project, other results were found that are a reason to question the equivalency factors calculated with USES. Normalisation data for Sweden were calculated making use of rather complete emission data from the Swedish EPA and extrapolated data from the extensive Dutch emission registration database (TUKKER and KLEIJN, 1996). On the Swedish total for human toxicity, chromium(VI) contributes 99%, chromium (III) 0.24% and lead 0.66%. This would imply that Sweden could reduce its human toxicity problems by 99.9% by concentrating its toxicity policy on just three substances, which seems a rather questionable result. A final, minor remark is that the equivalency factors for one of the PAHs, benz-a-pyrene (BaP), have been calculated making use of TDIs given by VERMEIRE et al. (1991). VERMEIRE calculated his TDIs for soil remediation situations, and stated explicitly that – in such situations – oral intake was considered as the only relevant uptake route for PAHs. PAHs present in air are mainly taken in by inhalation, which is a much more sensitive intake route due to carcinogenic effects on lung tissue. As a consequence, the TDI for the inhalation of BaP is a few 1,000 times lower than for oral intake. The LCA-USES manual thus greatly underestimates the relevance of BaP for human toxicity.

3. As shown by e.g. COWAN et al. (1995), probably the steady-state concentrations in the most important media such as water, soil and air can still be predicted with an acceptable uncertainty. But predicting the bioconcentration processes etc. in food webs, that specifically for persistent substances will be relevant for the potential toxic effects, seems to be of a different order of complexity (IJC, 1993).

4. In fact, authors like WYNNE (1987; 1993) mention the use of methodologies such as risk-assessment in different contexts as the context in which they were developed as an important reason for the creation of – what he calls – structural uncertainty.

5. During some lectures we held on this case, some people in the audience wondered if we had not applied the method to a rather difficult case. This suggestion does not seem to be appropriate. The draft LCIA method had been presented in a workshop in The Hague in April 1996. There, the main point of discussion was whether it would be appropriate to apply USES to heavy metals, since USES was originally designed for organic substances. In our case, we faced problems with straightforward and well-known organic substances such as phthalates, VCM, and EDC. Neither in the manual nor during that workshop were these substance groups specifically addressed as problematic. No one in my TNO/ CML project team foresaw the problems we finally faced.

5a. I submitted the initial draft of this paper in August 1997. In this final version, I only included the remarks made during the review process and did not try to include new developments in the debate. However, one remark has to be added here. Since late 1997, the attention for paradigmatic uncertainty seems to be enhanced in the LCA-community. Documents now produced in the framework of CHAINET and the Dutch Ecoindicator '98 project now refer to work of Thompson (e.g. THOMPSON, 1991) and Rotmans (ROTMANS and DE VRIES, 1997) as approaches to deal with (what I called: paradigmatic) subjectivity.

6. To give an example related to daily life: it is like not having a map of Amsterdam during the 1997 visit of the SETAC Europe conference, and using the map of Taorima you still had from 1996 instead, since that is the only other map you have – where just keeping your eyes open would bring you further.

7. An example of such an approach, where information on rather robust emission data is combined with rudimentary logic in order to set priorities, can be found in FINNVEDEN (1997). In a study performed by GÜNTHER and LANGOWSKI (1997) on resilient floor coverings, he noted that toxicity aspects had not been included, among others due to the fact that the authors were well aware of the problems related to the LCIA of toxic releases. FINNVEDEN argues, nevertheless, that a preliminary comparison of the toxicological impacts of PVC flooring and polyolefin flooring is possible. His argument is that both floorings are based on petrochemical resources, so that the potential impacts caused by the production of polyolefins will also be present in the life cycle of PVC. But, he argues, PVC has the additional drawback that there are potential impacts and uncertainties related to the production of chlorine (mercury emissions), the production of EDC and VCM (organochlorine emissions), and the use of additives (plasticiser emissions). Thus, in his view, it is reasonable to assume that polyolefins may score relatively better on toxicity impacts. In my view, the approach of building the argument is a good one, though in this specific comparison one point may have been forgotten. FINNVEDEN's argument only holds if the use of petrochemical resources for both PVC flooring and polyolefin flooring is about equal, and there are no other product chains with potential toxic impacts contributing to the production of polyolefin flooring. In that case, in the production chain of PVC flooring simply a number of additional emissions of potentially toxic substances can be found. However, in most cases the situation is more complex. In the production of pipes, for instance, considerably more petrochemical resources are needed for PE pipes as for PVC pipes. Thus, here one enters the difficult comparison of additional emissions in the production chain of ethylene (when PE is produced) with additional emissions specific for PVC.

8. Note that we thus reject an approach of "Anything goes", or an approach that allows that consensus can overrule robust scientific knowledge such as Ohm's law. An evaluative paradigm that, for instance, is based on the premise that mankind descends from cockroaches cannot, of course, be taken into account. We merely assume that in cases such as the LCIA of toxic releases the number of such robust scientific claims is insufficient for conclusive statements. The robust claims can be seen as a limited number of spots on a piece of paper, which still can be linked in a number of ways to draw a full picture.

9. In the near future, HOFSTETTER (forthcoming) will publish a Ph.D. thesis that uses and elaborates the ideas of THOMPSON et al. and ROTMANS and DE VRIES for LCIA. My own thesis analyses in depth the evaluative paradigms in the Dutch chlorine- and Swedish PVC-debate and, on that basis, provides a rough outlook of what such paradigm-dependent indicators should take into account (TUKKER, forthcoming).

   A main warning in my thesis is that a straightforward use of Thompson's cultural theory once again may easily lead to technocratic rules. The paradigms used by societal groups should be analysed as a start. For instance, the cultural-theory related indicator systems I have seen so far all miss one element that currently is the key topic in the Dutch chlorine debate. In concerns the possible release of chlorinated micropollutants, i.e. small but potentially toxic emissions of compounds not covered by classical monitoring systems.

10. By the way, some sociologists of science do not see Weinberg's idea as a way out of the problem, and even do not see a problem as such. They simply argue that all knowledge is constructed, that not any piece of knowledge is constrained by the real world, and that construction processes just should go ahead – the aim should be just to reach robust outcomes that are acceptable for societal actors (RIP, 1992), or are supported by the most powerful networks of scientists, politicians, etc. (LATOUR, 1987). After the expansionistic and reductionistic views, this would result in the relativistic view on LCA – or Machiavellistic, as some prefer to term it (AMSTERDAMSKA and HAGENDIJK, 1990), since indeed, in this view, power play is in fact the only thing that still counts.

11. In our view, reducing uncertainty by agreement on a convention after a consensus process or even a process backed by an authoritative panel is no solution. Such approaches lead inevitably to a construct that will not survive in a real controversy. Courts of Justice, for example, have decided not to accept even laws that prescribed certain sampling procedures for hazardous waste, since they gave no insight in the uncertainties.

12. In this respect, its seems useful to add one of the statements made by V.D. SLUIJS (1997) in his thesis: "For the main part, it will remain impossible to model the system earth".